esa

ECOSPHERE

SYNTHESIS & INTEGRATION

A four-component classification of uncertainties in biological invasions: implications for management

G. Latombe,[1,2,]† S. Canavan,[1,3] H. Hirsch,[1] C. Hui,[2,4] S. Kumschick,[1,3] M. M. Nsikani,[1] L. J. Potgieter,[1] T. B. Robinson,[1] W.-C. Saul,[1,2] S. C. Turner,[1] J. R. U. Wilson,[1,3] F. A. Yannelli,[1] and D. M. Richardson[1]

[1]*Department of Botany and Zoology, Centre for Invasion Biology, Stellenbosch University, Stellenbosch 7602 South Africa*
[2]*Department of Mathematical Sciences, Centre for Invasion Biology, Stellenbosch University, Stellenbosch 7602 South Africa*
[3]*Kirstenbosch Research Centre, South African National Biodiversity Institute, Private Bag X7, Claremont 7735 South Africa*
[4]*Mathematical and Physical Biosciences, African Institute for Mathematical Sciences, Cape Town 7945 South Africa*

Citation: Latombe, G., S. Canavan, H. Hirsch, C. Hui, S. Kumschick, M. M. Nsikani, L. J. Potgieter, T. B. Robinson, W.-C. Saul, S. C. Turner, J. R. U. Wilson, F. A. Yannelli, and D. M. Richardson. 2019. A four-component classification of uncertainties in biological invasions: implications for management. Ecosphere 10(4):e02669. 10.1002/ecs2.2669

Abstract. Although uncertainty is an integral part of any science, it raises doubts in public perception about scientific evidence, is exploited by denialists, and therefore potentially hinders the implementation of management actions. As a relatively young field of study, invasion science contains many uncertainties. This may explain why, despite international policies aimed at mitigating biological invasions, the implementation of national- and regional-scale measures to prevent or control alien species has done little to slow the increase in extent of invasions and the magnitude of impacts. Uncertainty is therefore a critical aspect of invasion science that should be addressed to enable the field to progress further. To improve how uncertainties in invasion science are captured and characterized, we propose a framework, which is also applicable to other applied research fields such as climate and conservation science, divided into four components: the need (1) to clearly *circumscribe* the phenomenon, (2) to measure and provide evidence for the phenomenon (i.e., *confirmation*), (3) to understand the mechanisms that *cause* the phenomenon, and (4) to understand the mechanisms through which the phenomenon results in *consequences*. We link these issues to three major types of uncertainty: linguistic, psychological, and epistemic. The application of this framework shows that the four components tend to be characterized by different types of uncertainty in invasion science. We explain how these uncertainties can be detrimental to the implementation of management measures and propose ways to reduce them. Since biological invasions are increasingly tightly embedded in complex socio-ecological systems, many problems associated with these uncertainties have convoluted solutions. They demand the consensus of many stakeholders to define and frame the dimensions of the phenomenon, and to decide on appropriate actions. While many of the uncertainties cannot be eliminated completely, we believe that using this framework to explicitly identify and communicate them will help to improve collaboration between researchers and managers, increase scientific, political, and public support for invasion research, and provide a stronger foundation for sustainable management strategies.

Key words: impact; invasive alien species; management; non-native species; uncertainty.

Received 2 November 2018; revised 15 February 2019; accepted 19 February 2019. Corresponding Editor: Debra P. C. Peters.

† **E-mail:** latombe.guillaume@gmail.com

Introduction

Biological invasions are a major threat to global biodiversity (CBD 2006). Accordingly, the Convention on Biodiversity dedicated Aichi Target 9 "to identify and prioritize species and introduction pathways, to control or eradicate priority species, and to put measures in place to manage pathways and prevent their introduction and establishment" (CBD 2010). Despite such international agreements, the implementation of national- and regional-scale measures required to prevent or control alien species is insufficient to prevent negative impacts from escalating (Early et al. 2016).

A major problem in invasion science is the proliferation of concepts and terms which are used in different ways by different parties. For example, the term "invasive species" is used to refer to several different concepts, and there is inconsistency in the way that scientists, policy makers, and managers apply the term (Heger et al. 2013, Courchamp et al. 2017, Sagoff 2018). The existence of alien and invasive species (however, they are defined) has been recognized and documented over several centuries (Darwin 1859). Nevertheless, only recently have consistent metrics and indicators been proposed to quantify biological invasions and their consequences, and impacts are still notoriously poorly documented (Simberloff et al. 2013). The mechanisms and values underlying biological invasion and their impacts can be highly context dependent, which hampers our ability to make useful generalizations. Such fundamental sources of uncertainty partly explain why, over the past 30 yr, the number of articles denying or trivializing the impacts of alien species, and indeed the field of invasion biology, has increased exponentially (Ricciardi and Ryan 2018, but see Munro et al. 2019). This has given rise to heated debates in both the mainstream media (Goode 2016, Rodriguez McRobbie 2016) and the scientific literature (Crowley et al. 2017, Davis and Chew 2017, Russell and Blackburn 2017*a*, *b*, Tassin et al. 2017, Guiaşu and Tindale 2018). Regardless of the motives of the criticisms (Duffy 2013, Collomb 2014), it is important to address conflicting views in a constructive fashion (Boltovskoy et al. 2018), and most of the issues raised have been repeatedly rebutted based on scientific evidence (Richardson and Ricciardi 2013, Russell and Blackburn 2017*a*, Ricciardi and Ryan 2018). Nonetheless, these criticisms highlight several key problems of coherence, ambiguity, or lack of agreement even among invasion scientists. A search in Web of Science using the terms "TOPIC: (uncertaint*) AND TOPIC: (invasive OR alien) AND TOPIC: (species)" returns 658 articles, 47% of them published in the last five years, showing that the importance of uncertainty is increasingly acknowledged in the literature on biological invasions. In our opinion, such uncertainties do not undermine the relevance of the field, as has been suggested by some authors (Valéry et al. 2013). Rather, they are a consequence of invasion science having undergone rapid growth recently and because the field is inextricably linked with many other disciplines from which it has borrowed concepts and terms (Vaz et al. 2017). Such problems of coherence must therefore be identified, elucidated, communicated, and, where possible, resolved, to improve the evidence-based foundation of invasion science in general and invasion management in particular.

A Four-Component Framework for Identifying Uncertainties in Invasion Science

To identify key uncertainties in invasion science, we propose a framework comprising four components: (1) the need to clearly circumscribe the phenomenon under study (i.e., the displacement of species beyond their native range); (2) the need to be able to quantify and therefore to provide evidence of the phenomenon (i.e., confirmation); (3) the need to understand the mechanisms that cause the phenomenon; and (4) the need to understand the mechanisms through which the phenomenon has consequences. Each of these components can be linked to specific sources of uncertainty, as we will show in the next sections. We argue that an applied scientific field where the circumscription, confirmation, causes, and consequences of key phenomena are well defined (or whose related uncertainties are at least clearly identified and communicated) is likely to progress faster toward the effective implementation of management measures. This is also important to ensure sustained scientific, political, and public support for actions, as was the case for climate and conservation science, as we discuss below. This is because identifying what phenomenon must be managed (circumscription and confirmation) and why (consequences) and how (causes) it should be managed will provide vital background for stakeholders and decision-makers (Fig. 1).

Taking climate change as an example, we suggest that the broad recognition of its existence (despite a small but highly vocal anti-climate change lobby) and the implementation of multiple and clear mitigation strategies that have been adopted by different actors at distinct scales to attenuate its effects and to stabilize its progression (IPCC 2014) can be explained by our framework: (1) There is a clear and easily understandable way of circumscribing the phenomenon using well-defined measurement variables (e.g., changes in global and local surface temperature; IPCC 2014); (2) the existence of the climate change is confirmed by consistent evidence obtained through extensive temperature records over wide spatial and temporal scales, based on robust methods of measurement (Peterson and Vose 1997); (3) a mechanistic understanding of its causes has been achieved (e.g., using mathematical models based on known bio-geochemical processes to represent the relationships between concentrations of greenhouse gases (especially CO_2) and temperature, and between greenhouse gas emissions and human activities; IPCC 2014), while also explicitly presenting existing uncertainties using a range of possible scenarios; and (4) the methods used to predict the consequences of climate change for the environment, biodiversity, and human welfare are being improved (Worm et al. 2006, Cardinale et al. 2012). Because the four components we identified above (circumscription, confirmation, causes, and consequences) are compellingly addressed in the case of climate change research, measures such as the Paris Agreement (United Nations 2015) have been achieved, which represents a necessary step toward mitigating the effects of climate change (though huge challenges remain, as shown by the

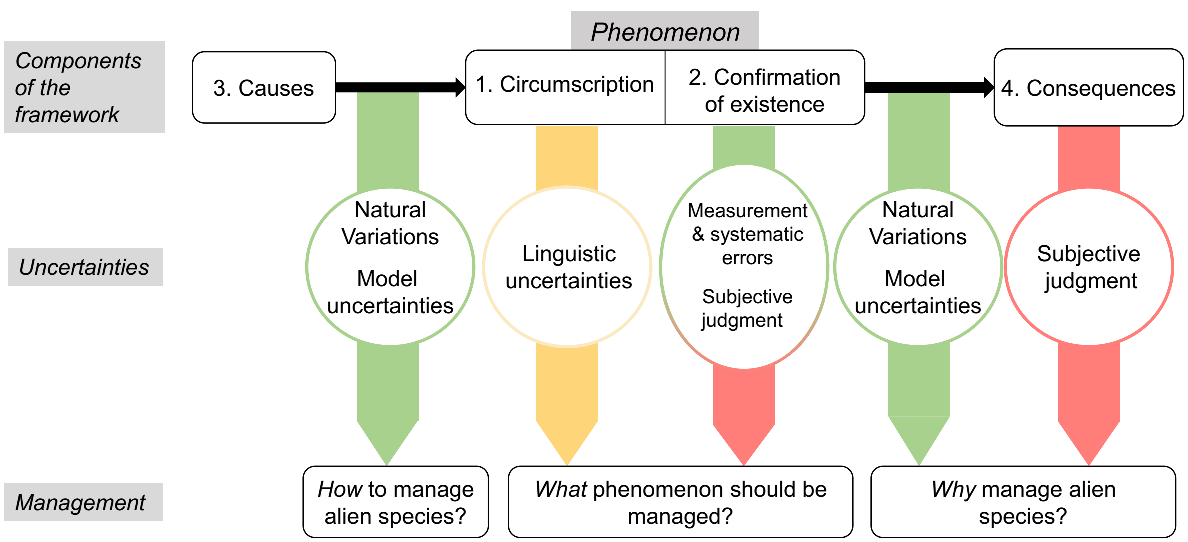

Fig. 1. The four-component framework (circumscription of the phenomenon, confirmation of the existence of the phenomenon, mechanistic causes of the phenomenon, and mechanistic consequences) upon which sustainable management actions must be built. These components are subject to specific types of uncertainties (linguistic in yellow, psychological in red, and epistemic in green) which can hinder the implementation of management actions and policies (see Table 1 for details). The natural variation uncertainties concern the mechanisms which result in alien species establishing and spreading, and through which alien species generate an impact, and therefore originate from the arrows. The other uncertainties concern the components themselves and therefore originate directly from them.

Box 1.

The 11 Types of Uncertainties Defined by Regan et al. (2002)

The 11 types of uncertainty are divided into epistemic, psychological, and linguistic categories. The psychological uncertainty was added to the original classification of Regan et al. because it differs from the epistemic uncertainties in that it is independent of the available technological, mathematical, or statistical developments, and can be resolved if properly acknowledged. See Table 1 for specific examples related to invasion science. Asterisks mark types of uncertainties that are explicitly discussed in this article.

Epistemic uncertainties

***Measurement error:** Imperfections in measuring equipment or observational technique generate a level of random variation in the data. Here, we include uncertainty resulting from the impossibility or extreme difficulty given the current technologies and techniques, to measure some aspects of the system of interest (in the extreme case, this would result in complete random variation in the data).

***Systematic error:** Imperfections in measuring equipment or sampling procedure generate a constant bias in the data.

***Natural variation:** Variations in the study system create differences between data collected at a specific location and time and the system considered as a whole (e.g., population variations in time). This uncertainty therefore results from incomplete information about the studied system; such uncertainty affects all fields of science. Without natural variation, the knowledge of a system is perfect, and there is no need to study it further. Natural variation leads to model uncertainty.

Inherent randomness: As indicated by its name, this type of uncertainty considers that the system is irreducible to a deterministic representation, even with complete knowledge of its elements. This kind of uncertainty is probably quasi-inexistent in real systems except for quantum physics, but the distinction is important to differentiate uncertainty resulting from imperfect information on the system.

***Model uncertainty:** This type of uncertainty arises from the necessary simplifications inherent in any conceptual or analytical model, for example, relating to the number of variables considered and their assumed causal or mathematical relationships. Numerical approximations are part of model uncertainty.

Psychological uncertainty

***Subjective judgment:** Uncertainty arising from the subjective interpretation of data, such as expert opinion.

Linguistic uncertainties

***Vagueness:** Lack of term for describing specific situations, especially those sitting at the border between existing definitions.

***Context dependence:** Uncertainty arising from the lack of specification of the context in which a given proposition must be understood. For example, the appreciation of height (as small or tall) would depend on whether we refer to an insect or a tree.

***Ambiguity:** The variety of definitions for a given word leads to such uncertainty.

***Underspecificity:** Lack of precision preventing from reaching clear conclusions from a proposition. Underspecificity can arise from lack of data.

Indeterminacy of theoretical terms: Uncertainty arising from an increase in knowledge, which makes previously used terms inadequate.

withdrawal from the agreement of some major countries).

A similar case regarding the four components can be made for conservation science, where some of the lessons learned from climate change science have been applied. Given the success of the Intergovernmental Panel on Climate Change (IPCC) at generating new science-based policies to mitigate climate change (IPCC 2014), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was initially modeled on the IPCC with the aim of mitigating biodiversity loss (Brooks et al. 2014). Other initiatives have been taken to mimic effective approaches from climate change mitigation, such as the definition of Essential Biodiversity Variables (Pereira et al. 2013), the minimum set of variables required to describe changes in biodiversity, modeled after the Essential Climate Variables (Bojinski et al. 2014). Tools applicable across a wide range of taxa have been implemented in many countries to mitigate biodiversity loss, such as biodiversity offsets, although their efficacy still needs to be improved (McKenney and Kiesecker 2010, Bull et al. 2013). The development of such national and international approaches to mitigate biodiversity loss is underpinned by a clear and unambiguous framing of at least three out of the four above-mentioned components: (1) The species concept as the unit used to measure biodiversity change is reasonably well defined, as are associated measures, such as species richness and phylogenetic diversity, allowing for a clear circumscription of the phenomenon; (2) the number and abundance of species is being constantly monitored and there is good evidence for the increase in species extinctions and decrease in species abundances (IUCN 2017*b*, World Wide Fund for Nature 2018); and (3) the major causes of species extinctions are reasonably well known and understood, with human-mediated habitat loss being the main driver (Sala et al. 2000). In contrast, regarding the fourth component, although there is clear evidence of the effects of biodiversity loss on ecosystem services such as on pollination, food provision, and water quality, among others (Worm et al. 2006, Hooper et al. 2012), the mechanistic understanding of these consequences is still fragmentary (Cardinale et al. 2012). Moreover, in addition to these knowledge gaps, different views exist on why and to what end biodiversity should be managed, depending on different values attributed to nature. Differences in value systems can lead to very different conclusions regarding the rationale and goals of conservation (Wallach et al. 2018, Driscoll and Watson 2019, Munro et al. 2019).

In what follows, we use the four-component framework for identifying, communicating, and ultimately reducing existing uncertainties in invasion science (Fig. 1). In doing so, we recognize epistemic, psychological, and linguistic uncertainties, based on Regan et al.'s (2002) classification (Box 1; Table 1). We show how these different types of uncertainties are detrimental for the implementation of management activities, how specific types of uncertainty correspond to each of the four components, and that their reduction requires different approaches, for which we propose solutions.

First Component: Circumscription of the Phenomenon

The need for clear and precise definitions

The first condition to measure and manage a phenomenon is to be able to circumscribe it clearly and unambiguously. The phenomenon of biological invasions is mostly circumscribed as species that are found outside of their native range due to the actions of humans (direct and indirect; Essl et al. 2018), and which are interchangeably termed alien, non-native, or exotic (Richardson et al. 2011). However, two main definitions of "invasive" are common in the literature, which introduces ambiguity, a form of linguistic uncertainty (Box 1). The first definition is based on the biogeographic invasion process, which is conceptualized as an introduction–naturalization–invasion (INI) continuum (Kruger et al. 1986, Williamson 1996, Richardson et al. 2000, Colautti and MacIsaac 2004, Richardson and Pyšek 2006, Blackburn et al. 2011, Robinson et al. 2016), and therefore contributes to circumscribe the phenomenon. The second definition is an extension of the first, but additionally requires alien species to have both spread and caused impact outside their native range to be called invasive. This definition is therefore also related to the consequences of the phenomenon (see section on component 4 below). The second definition is, for example, used by the Convention on Biological Diversity (CBD 2006), the United States Department of Agriculture (ISAC 2006), and the IUCN (https://www.iucn.org/theme/species/our-work/invasive-species). Confusing the issue further, invasive is sometimes used for native species that have recently increased in

Table 1. Summary of the four components of the framework (circumscription, confirmation, causes, and consequences) for the phenomenon of biological invasions.

Components	Definitions and examples for biological invasions	Associated uncertainties	Uncertainty class	Implications for management	Proposed solution
(1) Circumscription of the phenomenon	Species transported out of their native range by human activity, classified along the introduction–naturalization–invasion (INI) continuum	Ambiguity: The term "invasive" can have different meanings, based on biogeography, impact, or dominance; different definitions are adopted by different bodies	Linguistic	Public, policy makers, and scientists can be talking at cross-purposes (e.g., weeds [native or alien], invasive alien weeds, and invasive alien plants)	Indicate precisely which definition is used in a given study or context and link to existing frameworks. Avoid making imprecise analogies between different definitions; for example, avoid the term "native invaders"
		Underspecificity: Invasiveness defined in the biogeographic sense using the INI continuum does not acknowledge that different paths (increasing local abundance before spread, or the opposite) can lead to commonness	Linguistic	Invasions (regardless of the definitions used) arise through different processes and so there is no single management approach that suits all contexts	Species-specific management strategies should match the trajectory of the species: How frequent and how spatially targeted should the management actions take place?
		Vagueness: Some species are transported to novel environments through indirect human-mediated actions and lie at the interface between invasion and natural colonization	Linguistic	May lead to target incorrect species for monitoring/ underestimating invasion risk	The traits of the dispersal event may be more informative than classifying it as natural or human-mediated
(2) Confirmation of the existence of the phenomenon	Species databases (e.g., GRIIS, GloNAF)	Measurement/ systematic error: Human and taxonomic error, as well as inadequate data resolution, can impair the identification of alien species	Epistemic	Efforts at monitoring/ analyses of invasion risk are based on the wrong species	Constantly curate datasets to eliminate errors
		Measurement error: Not all countries or regions have the resources needed to establish and maintain monitoring programs, preventing from capturing worldwide fine-scale, temporal information on the state of biological invasions	Epistemic	The lack of standardized framework for acquiring data impairs the ability to assess the state of invasion across regions and taxa. Moreover, temporal information, although costly to obtain, is necessary to track the advancement of biological	Use a modular framework that enables countries to contribute to a global dataset, in accordance with their level of resources, with fine spatial- and temporal-scale data as the optimal situation

(Table 1. Continued.)

Components	Definitions and examples for biological invasions	Associated uncertainties	Uncertainty class	Implications for management	Proposed solution
				invasions and to assess the efficacy of management actions	
		Subjective judgment: The human perception of biological systems, and therefore of what makes a species alien, changes due to memory loss, that is, the shifting baseline syndrome	Psychological	Inclusion/exclusion of some species in management actions	Use verified records to define whether a species is alien in a systematic fashion, and use stakeholder engagement in a way so that decisions explicitly acknowledge value systems
(3) Mechanistic causes	Pathways of introduction: increases and changes in propagule pressure related to human activities, such as trade	Natural variation: Pathways of species change rapidly, such that biosecurity measures, if they are in place, might be based on historical rather than contemporary patterns (i.e., biodiversity lags)	Epistemic	Some new pathways developing or changing in a way such that current interventions are no longer appropriate. Biosecurity in one country is often dependent on that in neighboring countries	Greater international collaboration between biosecurity agencies. Horizon scanning or similar exercises to identify future risks and explore potential management needs
	Species invasiveness: Some species characteristics enable them to progress more easily along the INI continuum in novel environments. Species invasiveness is not independent from ecosystem invasibility	Natural variation/ context dependence/model uncertainty: The number of species and the variety of environments around the globe can generate context dependence in the findings on observed biological invasions, making it difficult to draw generalities on such mechanisms, leading to model uncertainty	Epistemic	The lack of a clear understanding of the mechanistic causes of the spread of species hinders our capacity to design appropriate management actions to contain their spread	Uncertainty linked to natural variation is common to all sciences and should only encourage researchers to pursue their investigations and refine their models. Efforts should be brought to the identification of the specificities of each study, to avoid context dependence uncertainty. Focus on the appropriate level for generalizations (i.e., invasion syndromes). Specify the nature of any comparison or analysis of invasiveness such that possible biases or competing explanations are clear
	Ecosystem invasibility: Disturbance and human modifications in the environment can favor the introduction	Natural variation/ context dependence/model uncertainty: as above	Epistemic	As above	Among the different management options commonly used (mechanical removal, chemical control, and biological control), some have been shown to be more appropriate in some

(Table 1. Continued.)

Components	Definitions and examples for biological invasions	Associated uncertainties	Uncertainty class	Implications for management	Proposed solution
	and spread of alien species				contexts, with respect, for example, to the spatial scale of the receiving environment and the position of the alien species along the INI continuum. Such correlative relationships resulting from past trial and error experiments are still valuable while the mechanisms are investigated
(4) Mechanistic consequences	Impacts on the ecological and socioeconomic systems	Natural variation/ model uncertainty: The importance of different mechanisms to explain the impact of alien species is likely to vary with many factors, leading to model uncertainty. The relationship between the presence of alien species and some ecosystem services is still poorly known	Epistemic	The impact of alien species that have not spread in many environments is difficult to forecast in the absence of mechanistic understanding of the impacts. Ignorance of the impact mechanisms also makes the assessment of the impact category in the EICAT scheme problematic	Increase the number of studies of alien species and invaded environments and focus on efforts to collate and compare existing studies to reveal the basic mechanisms leading to deleterious impacts. Precautionary principles can be used, especially if similar species had large impacts somewhere in the past
		Subjective judgment/context dependence: A measure of impact necessarily addresses one specific aspect of the impact. Emphasizing one measure over another therefore reflects a value	Psychological	Not acknowledging the values linked to a specific scheme for deciding on management activities will create polarization and distrust between managers and the stakeholders	Rely on objective systems for measuring impact, such as EICAT and SEICAT, and be explicit to ensure transparency. However, do not rely only on impact or risk assessments for decision-making—rather, develop and implement risk-analysis frameworks; these require contextual information on additional aspects, such as management feasibility and costs
		Subjective judgment: Some alien species have a positive ecological or socioeconomic impact	Psychological	As above	Involve stakeholders with different interests (positive and negative) in studies and management activities

Notes: For each component, the corresponding uncertainties (belonging to different classes; see Box 1) are detailed, which can have implications for management. Possible solutions to solve these uncertainties are suggested. EICAT, Environmental Impact Classification for Alien Taxa; SEICAT, Socio-Economic Impact Classification for Alien Taxa.

range or abundance in a manner that has undesired impacts. The use of the phrase "native invasions" in this context (Nackley et al. 2017) is unfortunate as it conflates biogeographic, demographic, and impact phenomena.

In the INI continuum, alien species are characterized as invasive if they establish, reproduce regularly, and spread over substantial distances in the new environment. Alien species which only reach, survive, and occasionally reproduce in a novel environment are qualified as casual, and those that survive and reproduce but remain constrained to a specific location (i.e., do not disperse widely) are termed naturalized (Richardson and Pyšek 2006, Blackburn et al. 2011). It is notable that inherent differences between environments result in some terms being more readily used in terrestrial than aquatic systems. For example, casual is seldom applied to aquatic taxa (probably due to difficulties in directly observing organisms in marine and freshwater habitats), while naturalized is used in both systems (Richardson and Pyšek 2006, Robinson et al. 2016). The trajectory from low to high local abundance and from narrow to wide geographical range is nonetheless not unique. Four different demographic criteria are commonly used to describe alien species and commonness (local abundance, geographic range, environmental range, and spread rate; Catford et al. 2016), and the path to commonness can progress differently along these characteristics for different species (McGeoch and Latombe 2016). Not acknowledging this complexity can create uncertainty through oversimplification, that is, underspecificity (Box 1). Note also that the notions of substantial distance, wide dispersal, and wide geographic range can themselves be vague (Box 1). These notions can vary depending on the species and life-form of interest and can also vary between ecosystem types. For example, dispersal in insects can occur at various scales, and movement with potential consequences for gene flow can differ from routine movement or migration (Renault et al. 2018). In this instance, the relevance of movement is defined functionally rather than by distance. Mode of dispersal can also affect the geographic scale at which spread occurs: the spread of alien taxa with limited dispersal capabilities (e.g., live-bearing anemones) occurring over smaller geographic scales as opposed to the spread of highly mobile species (e.g., European starlings) or those that employ broadcast spawning (e.g., mussels; Branch and Steffani 2004, Berthouly-Salazar et al. 2013, Robinson and Swart 2015). Knowledge of the biology of the species of interest is crucial for resolving this type of uncertainty.

As mentioned above, the commonly accepted definition of an alien species used to circumscribe the phenomenon of biological invasions is a species that has been introduced outside of its native range due to human actions (CBD 2006, Richardson and Pyšek 2006, Blackburn et al. 2011, IUCN 2017*a*), regardless of whether introductions were intentional or accidental (Hulme et al. 2008, Hulme 2009, Wilson et al. 2009). Natural colonization—the range expansion of species without direct intervention of humans to aid dispersal (Russell and Blackburn 2017*b*)—is similar to human-mediated invasions in that both types of movement involve the negotiation of barriers. However, the two phenomena differ qualitatively (e.g., pathways) and quantitatively (e.g., frequency, with the rate and extent of human-mediated dispersal of species being at an unprecedented level and over long distances compared to natural colonization); this has important implications for policy and management (Wilson et al. 2016). The distinction between natural and human-mediated range expansion is especially significant in marine environments where range expansions occur at least an order of magnitude faster than in terrestrial systems (Sorte et al. 2010), suggesting that the separation between human-mediated translocations and natural colonization might be greater in terrestrial settings, at least with regard to rate of spread. Nonetheless, some range expansion mechanisms lie at the interface between the two concepts of natural colonization and human-mediated invasions, making such a discrete distinction problematic, and generating vagueness (Box 1). Such expansion mechanisms include tsunamis that transport anthropogenic debris carrying species, range shifts due to human-mediated climate change, and range shifts due to human-mediated modifications to biogeographic barriers, such as the Lessepsian migration following the opening of the Suez Canal (Hoffmann and Courchamp 2016, Nackley et al. 2017). In response to this, protocols have been developed that explicitly incorporate estimates of uncertainty (Essl et al. 2018).

Implications for management and solutions

All the uncertainties related to the first component of the framework belong to the linguistic class (Box 1). From a rational perspective, they should be the easiest types of uncertainty to resolve, since solutions need not rely on technological, mathematical, or statistical advancements (Box 1, Table 1). However, they can also be extremely detrimental to the field and to the implementation of management actions (Fig. 1), as they generate confusion and create the impression of a lack of coherence. This is especially true for the ambiguity resulting from the multiple meanings attached to the term invasive which can thwart clear engagement with policy makers, stakeholders, and the public. Our objective here is not to prescribe what invasive should mean, nor to provide a terminology for invasion science, since this has been done elsewhere (Richardson et al. 2011, Hui and Richardson 2017). Biological invasions can be perceived and defined differently depending on the research or management context, and trying to impose a unique definition is counterproductive (Heger et al. 2013). Scientists must, however, be precise in the definition they are applying in a given context and must relate the chosen definition to existing frameworks (Robinson et al. 2016, Courchamp et al. 2017). As such, in the context of the four components presented here, invasive refers to the biogeographic definition provided by the INI continuum, since it is required to obtain a precise circumscription of the phenomenon and to increase coherence between the four components of the framework.

A better understanding of how different paths to commonness (based on changes in local abundance, geographical range, and spread rate) result in invasions can assist with management planning. Species that become locally common before spreading require more intensive local actions, whereas species undergoing long-distance dispersal may necessitate management actions over large areas. In specific contexts, especially at small spatial scales, such considerations for reducing underspecificity of the terminology may therefore be used to complement the INI continuum and improve management actions and are especially vital for a mechanistic understanding of invasions (see component 3).

The vagueness emerging from the simplification of terminology linked to the need to classify species into discrete classes can result in the exclusion of species that are potentially able to cause impacts in the receiving environment from management practices (see component 4 below for more details on impact). In a similar manner to understanding the paths to commonness, Wilson et al. (2009) suggested that when looking at dispersal, it is important to look explicitly at the traits of dispersal rather than just categorizing the pathways.

Second Component: Confirmation of the Invasion Phenomenon

Uncertainties linked to available data and monitoring

Many datasets of alien species distribution are now available at different spatial scales, which makes it easier to confirm the phenomenon (GRIIS; Pagad et al. 2015). However, datasets are often spatially heterogeneous, and this heterogeneity can vary across life-forms (van Wilgen et al. 2018). Additionally, many countries have limited resources and have not invested in detailed monitoring of alien species (Latombe et al. 2017). There is also no consensus on the type of classification schemes used to compile datasets with regard to differentiating between invasion stages (see section *First Component: Circumscription of the Phenomenon*) but also with regard to taxonomy. Lastly, few datasets (but see, e.g., Dyer et al. 2017, Dornelas et al. 2018) provide the temporal information on the occurrence of alien species required to track the increases in the spread of biological invasions (Seebens et al. 2017).

Leaving aside ambiguities related to the definitions of invasiveness described above, determining the status of a species as native or alien is not straightforward (Latombe et al. 2017, Essl et al. 2018). This complicates the creation and use of datasets such as those mentioned above. Sources of error have both human and taxonomic dimensions, and inadequate data resolution or availability are also problematic (i.e., measurement errors and systematic errors; Box 1; see McGeoch et al. 2012 for a detailed list of such uncertainties). Determining the status of species as alien or native is especially problematic in the marine realm and for microbial and soil organisms, in which cases long-term historical records are scarce and many species are cryptogenic (Ojaveer et al. 2018). Furthermore, since humans have relocated species as they have moved and traded around the globe for millennia (Hulme 2009), different alien species at the same location are likely to have different residence times. For example, a classification between archaeophyte (arrived before 1500 AD) and neophyte species (arrived after 1500 AD) has been widely used in Europe (Pyšek et al. 2003, Hui et al. 2013, Latombe et al. 2018), but it is a much less clear-cut distinction for other continents. Public perceptions of what constitutes an alien species are also important, and such views are subject to the shifting baseline syndrome which posits that the human perception of biological systems (and their original species composition) changes due to memory loss (Papworth et al. 2009), a phenomenon also termed ecological amnesia (Seidensticker 2008). Such changes (or, the opposite, inertia) in the local and global socio-ecological memory are linked to different pathways of developments and to existing natural resource management practices and are highly complex (Nykvist and von Heland 2014). For example, the shifting baseline syndrome can accelerate through positive feedbacks as poor management practice will result in degraded environments and their acceptance as the norm, which further promotes poor management practices (Soga and Gaston 2018). This memory loss therefore involves subjective judgment (Box 1).

Implications for management and solutions

Having accurate assessments of the state of invasions is an obvious requirement for implementing management actions (Fig. 1). Also, having the capacity to compare information from different locations (countries or regions) and from different datasets is necessary to obtain a global picture of the state of invasions. Temporal information is extremely valuable, not only for deciding on management strategies, but also to assess the efficacy of existing ones. More effort should be directed toward acquiring such temporal data at various spatial scales, updating existing databases, and making the data freely available online. Errors of identification of species can lead to both under- and overestimation of the state of invasions in a region or country. Datasets should thus be curated in perpetuity if errors are to be minimized.

Not all countries or regions have the same level of resources for monitoring the introduction and spread of alien species (Early et al. 2016). As for species monitoring programs for conservation purposes (Schmeller et al. 2015), or more generally for the monitoring of global changes in the biosphere (Running et al. 1999), a standardized, modular framework in which all countries can contribute at the level of their capacities and strategically contribute to a global information set over time is required, and has been proposed to improve data comparability (Latombe et al. 2017). In this framework, monitoring can be as simple as establishing national lists of alien species delimited by country borders and can be extended to cover a set of priority sites. The latter can also be further developed to a national level to obtain information on the area occupied by species, until a network of long-term monitoring sites is established for acquiring temporal information. This framework still needs to be formally endorsed by international organizations, which would encourage its application worldwide. Rejmánek et al. (2005) identified the United Nations as the international organization appropriate for managing a global database of alien species. Smaller-scale resolutions may nonetheless be more efficient as an initial step. The European Union (EU) has developed a substantial body of environmental initiatives turned into laws that ensures coordination between countries at the continental scale (Ojaveer et al. 2018). Endorsement of such frameworks by the EU could represent a leading action toward global initiatives at the UN level. There is also a need for metrics and indicators linking the INI continuum to field observations and management and policy decisions (Wilson et al. 2018). Wilson et al. (2014) provide an example of how to measure tree invasions in practice to ensure the effective linkage to management and policy decisions. There is a need to develop more context-sensitive metric systems, like that already developed for trees (Wilson et al. 2014), to appropriately support management at various scales, among taxonomic groups, or between terrestrial and aquatic systems. Finally, citizen science offers the means to collect data and contribute to a surveillance network of monitoring (Pocock et al. 2018). In combination with proper education about past ecosystems (Soga and Gaston 2018), citizen science can also contribute to preventing the shifting baseline syndrome by making citizens more familiar with the natural environment.

Third Component: Mechanistic Understanding of the Spread of Alien Species (Causes)

Three main factors influence the spread of alien species in a given environment (Catford et al. 2009): (1) Pathways of species introduction created by human activities enable the species to reach the new environment; its spread will then depend on the interdependent characteristics of (2) the alien species (invasiveness) and (3) the receiving socio-ecological environment (invasibility).

Pathways of species introductions

Alien species are introduced to areas outside their native ranges through a multitude of pathways, whose relative importance depends on the characteristics of the organism (Hulme et al. 2008). There is extensive and growing literature on pathways, both from a general perspective (Hulme et al. 2008, Hulme 2009, Wilson et al. 2009) and for specific taxa, regions, and pathways (Reichard and White 2001, García-Berthou et al. 2005, Katsanevakis et al. 2013, Faulkner et al. 2016, Saul et al. 2017). The importance of pathways has long been recognized and their identification is explicitly required in Aichi Target 9 (CBD 2010). Furthermore, a pathway classification scheme has recently been endorsed by the UN (Scalera et al. 2016). However, substantial uncertainties are still associated with the delineation of pathways. Data on the propagule pressure associated with different pathways are still scarce (Pyšek and Richardson 2010), and propagule pressure of unintentionally introduced species is difficult to quantify. This is particularly relevant for marine systems, microbial organisms, and soil ecosystems, where few introductions occur intentionally (Williams et al. 2013) and direct observations of propagules are challenging. Moreover, pathways for species change in time, and their importance for introducing alien species to a region can vary substantially over time (Wilson et al. 2009). Such uncertainties fall under the umbrella of natural variations (Box 1; Table 1).

Species invasiveness and ecosystem invasibility

Many abundant alien species have similar traits to the most abundant native species (Thompson et al. 1995, Meiners 2007, Leishman et al. 2010), a strategy that corresponds to joining the locals (Tecco et al. 2010). However, mechanisms relying on species difference (e.g., enemy release, prey naiveté, novel weapon) have been proposed to explain the success of some invasive species, described as trying harder (Tecco et al. 2010) to gain an advantage in the competition–colonization space (Levins and Culver 1971). The paradox between the two strategies (success based on traits that are similar or different from those of natives) has been acknowledged since Charles Darwin formulated what is now known as Darwin's naturalization conundrum in which species phylogeny is associated with their traits and invasion strategies (Cadotte et al. 2018). Generally speaking, invasion success may depend on the eco-evolutionary experience that introduced and native species share with each other (Saul et al. 2013, Saul and Jeschke 2015), but the relative importance of biotic mechanisms in this process is context dependent (e.g., terrestrial and marine systems; Papacostas et al. 2017), making it difficult to identify generalities (but see Hui and Richardson 2019). In addition, the introduction of some species can modify the receiving environment, making it more suitable for further invasions through positive feedbacks (Gaertner et al. 2014). For example, fire-tolerant alien plants can alter the fire regime of their environment by changing fuel properties (Brooks et al. 2004), while species tolerant of anti-fouling paints can facilitate invasion by other more sensitive taxa by offering a non-toxic settlement substrate (Floerl et al. 2004). Historical contingencies, such as residence times, can also hinder the detection of a signal of species traits affecting invasiveness (Wilson et al. 2007). Finally, the comparison of invasive and native species traits can be performed in many ways, which can generate different results, and such analyses depend on the question of interest (van Kleunen et al. 2010, Hulme and Bernard-Verdier 2018). These uncertainties therefore result from the complexity of the issue and fall under the umbrella of natural variations (Box 1; Table 1).

Independent disturbance and human modifications can also generate conditions which enhance the probability of alien species reaching and thriving in novel environments independently of their traits (MacDougall and Turkington 2005); that is, alien species are passengers, as opposed to drivers of change. For example, roads can increase propagule pressure due to increased human-facilitated dispersal (Spellerberg 1998), and increased nitrogen levels (e.g., from agricultural activities) can modify abiotic conditions to favor alien species to the detriment of native species (Brooks 2003). Soil erosion (sometimes due to other invasive species) and the consequent reduction of native species can decrease biotic competition, providing alien species opportunities to invade (Catford et al. 2012). Spatially synchronized environmental fluctuations can further reduce the temporal variations of regional invader populations and boost their viability, pointing at a scale-dependent invasiveness (Hui et al. 2017). There are still many uncertainties regarding these different mechanisms and their roles in mediating ecosystem invasibility. Again, these uncertainties result from the complexity of the issue and fall under the umbrella of natural variations (Box 1; Table 1).

Implications for management and solutions

Changes in the importance of pathways for introducing alien species and the potential lack of recent data on the associated propagule and colonization pressure can lead biosecurity measures to be based on historical rather than contemporary patterns. It is therefore necessary to identify the factors that promote biological invasions as new infrastructures linked to social and economic developments are created. For example, new practices such as blue-green infrastructures, intended to alleviate human pressure on ecosystems, can promote alien species introduction and spread if not done properly (Angelstam et al. 2017, Li et al. 2017); but considerations of the implications for biological invasions are often lacking in practice.

Once introduced, trial and error can help identify appropriate management actions for alien species, and successful management actions for one species in a given environment can serve as guidelines for similar species and environments. Yet this approach is usually case-specific and can be both inefficient and expensive. The most effective approach (e.g., mechanical, chemical, or biological) depends on different factors including the life-form of the species, the position of the species along the INI continuum, and the spatial scale at which the control action must be conducted (van Wilgen et al. 2000). General recommendations therefore require caution (Wittenberg and Cock 2001). A mechanistic understanding of the causes for invasion is necessary to provide a broader comprehension of the likely outcomes of management actions (Fig. 1; Hulme 2003). The current lack of understanding of such processes due to natural variations, and the entailed model uncertainties (Box 1), results from the relatively short time scientists have studied biological invasions as a global phenomenon. Current investigations of such processes should be maintained to improve our understanding of the invasion processes, and their potential to allow for generalizations to be made (i.e., invasion syndromes; Kueffer et al. 2013). This task will be further complicated by the fact that, in combination with other factors such as land use and climate change, biological invasions have steered many ecosystems away from historical trajectories. Such changes have led to so-called novel or hybrid ecosystems, for which new conservation approaches might be needed (Hobbs et al. 2014), further complicating the management of alien species.

Fourth Component: Mechanistic Understanding of the Impact of Alien Species (Consequences)

Sources of uncertainties regarding impact

Uncertainties linked to the impact of alien species are complex, as they originate from many sources and belong to two different categories: natural variations and subjective judgment (Box 1; Table 1). It is therefore necessary to detail the different sources of uncertainties before distinguishing between these two categories.

*Defining harm.—*As for the first component of the framework, the key condition for assessing the impact of alien species is a clear definition of such impact (Jeschke et al. 2014). The notion of harm used in the impact-based definition of an invasive species has been criticized as being based on value (i.e., subjective judgment; Box 1) and therefore not scientific (Sagoff 2009, 2018). Harm can nonetheless be objectively and scientifically defined and quantified as a negative variation of a precisely defined measure of impact based on a given referential. The Environmental Impact Classification for Alien Taxa (EICAT; Blackburn et al. 2014, Hawkins et al. 2015, IUCN 2017*a*, *b*) is a good example of such a quantification: It is a semiquantitative scheme that assesses the impact of the presence of an alien species on native communities by evaluating how it decreases native individual fitness (minor impact—harmful to native individuals), decreases native population densities (moderate impact—harmful to native populations), or causes reversible (major impact) or irreversible (massive impact) changes in native community composition (loss of native species, i.e., harmful to native species composition).

Like the EICAT scheme, the Socio-Economic Impact Classification for Alien Taxa (SEICAT; Bacher et al. 2018) quantifies the negative impact of alien species on human welfare as minimal, minor, moderate, major, and massive, by evaluating how difficult it is for people to perform their normal activities. As with EICAT, an alien species can have a negative impact on (i.e., be harmful to) human welfare without completely preventing activities, and a baseline of normal activities in the absence of alien species must therefore be defined (note that normal activities do not necessarily represent a desirable level, as additional factors, especially economic ones, would be required to do so).

Ecosystem valuation (Costanza et al. 1997, 2017) offers an approach for quantifying the impact of alien species using economic values. It has been applied to compute the economic cost (i.e., a negative impact, or harm) of alien species based on provisioning services, due to the lack of data availability for other services such as regulating and cultural ones (Vilà et al. 2010). In theory, ecosystem valuation techniques could also result in positive gains; unlike the two schemes mentioned above, it therefore has the advantage of enabling us to quantify potential positive effects (see section *Social, economic, or ecological benefits of alien species*) on native species richness which, despite detrimental effects on some native species or impacts of minimal concern, may result in a net gain in species richness. Ecosystem valuation can also account for different perspectives on the value of nature, including efficiency, fairness, and sustainability, using different methods specific to each perspective (Costanza and Folke 1997).

Pitfalls for quantifying negative impact.—Although rates of species extinction are of major interest, direct metrics of impact focusing on current rates only are problematic, especially when assessing impacts of invasions on native plants (Downey and Richardson 2016). Alien species can modify species interactions (Papacostas et al. 2017) or the environment (Bax et al. 2003) in ways that might only lead to loss of biodiversity in the future (Essl et al. 2015). The Minor and Moderate impact categories of EICAT provide a possible solution to this issue, as they may lead to anticipate future Massive or Major impacts, but huge challenges remain to make such predictions, due to natural variations uncertainties (Box 1).

In addition, the difficulty of quantifying impact varies across environments. Although, unlike many other impact schemes, EICAT was designed to be practically applicable, and has been applied, across many taxa (Evans et al. 2016, Kumschick et al. 2017, Hagen and Kumschick 2018, Kesner and Kumschick 2018, Canavan et al. 2019*b*), its applicability for guiding management in a marine context has been questioned due to differences in knowledge on invasion records (Ojaveer et al. 2015). Despite a rebuttal to these critics (see the response by Blackburn et al. 2015 as a comment to Ojaveer et al. 2015), differences of knowledge between environments will indeed lead to differences in uncertainties on impact quantification, which also fall under the umbrella of natural variations (Box 1).

Social, economic, or ecological benefits of alien species.—Although the vast majority of studies on biological invasions have focused on the negative impacts (harm) of alien species, invasions can also have positive ecological (if they are beneficial to certain native species) and socioeconomic (if they cause an increase in the income and welfare of people) impacts. For example, several species of Californian butterflies have expanded their geographic range and their flight seasons by feeding on exotic plants (Graves and Shapiro 2003). In the southwestern

United States, *Tamarix* plant species from Eurasia provide habitat for native bird species, including rare and endangered ones (Sogge et al. 2008), while along the South African coast the spread of an invasive mussel has contributed to increased populations of the previously endangered endemic African black oystercatcher (*Haematopus moquini*; Branch and Steffani 2004).

From a socioeconomic perspective, many rural communities in South Africa make cultural and economic use of alien prickly pears (*Opuntia ficus-indica*) and black wattles (*Acacia mearnsii*; Shackleton et al. 2007, 2011, Beinart and Wotshela 2012), while the entire mussel and oyster culture industries in the country are based on the invasive Mediterranean mussel (*Mytilus galloprovincialis*) and the Japanese oyster (*Crassostrea gigas*; Olivier et al. 2013). Alien species are also often purposefully introduced to novel environments for their direct economic and social benefits. For instance, more than a hundred *Eucalyptus* species were introduced in South Africa to provide timber and are now serving as sources of nectar and pollen for honey production (Allsopp and Cherry 2004, Forsyth et al. 2004).

It can be very difficult, however, to determine whether such benefits balance out simultaneous detrimental ecological and socioeconomic effects. In contrast to the range expansion of some Californian butterflies mentioned above, at least three other species of butterflies have been observed to lay eggs in alien species that are toxic to larvae (Graves and Shapiro 2003). Some of the above-mentioned *Eucalyptus* species introduced in South Africa for timber production cause major disruptions to invaded ecosystems, particularly riparian habitats (Tererai et al. 2013), by using large amounts of water (Le Maitre et al. 2016). The rainbow trout (*Oncorhynchus mykiss*), native to the United States, was introduced to many countries for recreational fishing, and is viewed positively by recreational anglers, but has detrimental effects on native biodiversity (Shackleton et al. 2019). The same invasive mussel upon which the South African mussel industry is founded and which has benefitted African black oystercatchers has dramatically altered the structure of rocky shore communities in the region (Sadchatheeswaran et al. 2018).

Alien species can also simultaneously have both positive and negative socioeconomic impacts. *Echium plantagineum*, a forb native to Europe, produces abundant nectar with positive economic effects for beekeepers in Australia, but is toxic to livestock and therefore causes negative economic impacts for agriculture (Cullen and Delfosse 1984). In many cases, those stakeholders who benefit from alien species are not the same as those who bear the costs of the resulting invasions (they might live in different places or work in different sectors of the economy, or the benefits might be now and the costs only much later; van Wilgen and Richardson 2014).

*Negative impact of native species.—*As with invasive species, some native species can proliferate extensively and cause considerable changes to ecosystem processes (Simberloff et al. 2012, Nackley et al. 2017). It is therefore important to describe the differences between the impacts of these two categories of species, especially for designing management actions (see section *Implications for management and solutions*). Pivello et al. (2018) define these as super-dominant. Super-dominant species arguably receive less attention in the scientific literature despite the high levels of impact they cause because of their native status. However, the impact of native species is highly dependent on the intensity of human-induced disturbance (Simberloff et al. 2012, Canavan et al. 2019*a*). Natives are generally less likely to have widespread detrimental impacts on native communities than alien species, whose impacts can be high even in the absence of human disturbance (Paolucci et al. 2013, Hassan and Ricciardi 2014, Taylor et al. 2016). By conflating these super-dominant species with invasive species (i.e., vagueness; Box 1), the mechanistic consequences and appropriate management responses can be less clear. Nonetheless, as for invasions, it is important to circumscribe, confirm, and understand the causes and consequences of such super-dominant native species. Furthermore, at least in some cases, the impacts and management might be similar in both the native and alien ranges (Canavan et al. 2019*a*).

Implications for management and solutions

*Relationship between the origin, impact, and management of a species.—*It has been argued that considering the biogeographic origin of a species (i.e., whether the species is alien) as a basis for determining whether management is required is based on xenophobia, and that species should be controlled based on their impact, irrespective of whether they are native or alien (Davis et al. 2011, Valéry et al. 2013). We agree in that understanding the consequences of biological invasions is necessary to improve their management (Fig. 1) and that the negative impacts of native species call for their management. Yet, in view of the above-mentioned differences in impact between these two categories, we argue that being clear about the native–alien distinction will help reduce uncertainties or at least make them explicit. Similarly, the decision whether to consider the alien-native distinction in indices of biodiversity, sustainability, and global change (Schlaepfer 2018) will depend on the value system and the intended purpose of such indices. If the overall goal is to conserve biodiversity, then the loss of native species richness at local scales, the loss of overall species richness at a global scale, changes to beta-diversity through homogenization and other processes, and threats to ecosystem processes are undesirable. As a corollary, the default should be to treat alien and native species separately in indices of biodiversity (Pauchard et al. 2018). This is especially true as there are many uncertainties around the possible real impact of some alien species, either because they may have impacted native species in ways that have not been recorded yet (measurement errors; Box 1) or because their impact will only become apparent in many years. Including alien species as part of biodiversity on the same level as native species without discrimination is therefore risky, especially for species which have escaped their historical range recently. This approach could promote the dissemination and spread of harmful alien species and should be avoided. However, this decision might need to be modified on a case-by-case basis (see also section on *Management and values* below addressing the issue of conflict-generating species which have positive and negative impacts for different stakeholders).

Decisions on which alien species to manage, and when and how, do not depend only on the impact of such species. Other considerations include the value of the receiving environment (which can be assessed using different criteria, such as levels of native species endemism and recreational value) and the feasibility of species management. For example, species with little current impact and spread would probably be more easily managed than already naturalized species (Wittenberg and Cock 2001). Schemes like EICAT and SEICAT therefore are therefore not risk assessments and should ideally form one component of a broader impact assessment—in particular, such schemes should not be used in isolation to produce a statutory list of harmful alien species (IUCN 2017*a*, *b*).

The mechanisms explaining the impact of alien species, and the geographical and temporal scale at which they take place, are still incompletely understood and are likely to vary with the life-forms (e.g., animals vs. plants; trees vs. herbs) and the nature of the receiving environment (e.g., terrestrial vs. freshwater; lentic vs. lotic). More work is therefore required to fully understand their detailed nature (Colautti et al. 2004), and to reduce model uncertainties resulting from natural variation (Box 1; Table 1). Nonetheless, there is a general consensus that alien species are much more likely to have higher negative impacts than native species and are more likely to have negative than positive impacts on native biodiversity (Goodenough 2010), as discussed above. The various putative explanations for such impacts call for a precautionary principle and for their control, even in the absence of a clear mechanistic understanding of their impact. This is especially true for long-lived species such as trees, for which the impact of alien species on the native biodiversity is likely to only occur after lengthy time lags (Richardson and Ricciardi 2013, Richardson et al. 2015).

Management and values.—Choosing a strategy for managing alien species is not straightforward. Besides potential epistemic uncertainties (mentioned above) on which to base such a decision, choosing a management strategy is necessarily affected by a set of ethical and political values, as well as by people's perceptions which are influenced by multiple factors (Shackleton et al. 2019). Acknowledging such values is crucial to avoid adverse effects (Essl et al. 2017).

Deciding which measure of impact to use is not a trivial task. The two formal frameworks for measuring the impact of alien species across taxa described above measure impact on different personal, societal, or political values (i.e., subjective judgment; Box 1). Socio-Economic Impact Classification for Alien Taxa is an anthropocentric, "nature for people" perspective (Costanza et al. 1997, Kareiva and Marvier 2012), while EICAT aligns more with a "nature for itself" viewpoint (Soule 2014). A "nature and people" perspective (Mace 2014) would require using both schemes simultaneously (Hagen and Kumschick 2018). Furthermore, measures other than SEICAT may be required to best represent the value of different societies to define human welfare, for example, when weighing different economic and cultural impacts is required. This question is therefore more philosophical than scientific and is similar to the debate between the new conservation (Kareiva and Marvier 2012) advocates and their opponents in conservation science. It should nonetheless be explicitly recognized. The pending assessment of invasive alien species of IPBES (2018) acknowledges both ecological and socioeconomic impacts of alien species, and therefore provides an opportunity to design a robust methodology to account for such complexity.

"Conflict-generating species" (Dickie et al. 2014, Zengeya et al. 2017), which have both positive and negative ecological and socioeconomic impacts, will also be valued differently by the different stakeholders (i.e., subjective judgment; Box 1). It is therefore crucial to involve all stakeholders in the decision process leading to management activities (Novoa et al. 2018).

Final Considerations

Although humans have moved species around for millennia, the recent intensification in the forces driving biological invasions threatens both environments and human welfare. Managing alien species (which, where, when, and how) is an increasingly complex component of environmental management in the Anthropocene and faces many challenges. Biological invasions, climate change, and biodiversity loss are interdependent and tightly embedded in increasingly complex socio-ecological systems. An inevitable consequence of this is the emergence of increasingly complex wicked problems where the cause-and-effect relationships between components, be they logistical components or stakeholders involved in management, are unordered. Solutions to such problems are not always obvious and often demand the consensus of many stakeholders to define and frame the dimensions of the phenomenon, and to decide on appropriate actions (Woodford et al. 2016). Acknowledging, communicating, and, when possible, breaking the phenomenon down into bite-size components to reduce complexity is sometimes necessary for action. Caution is required however, as approaching this in a simplistic fashion, ignoring the uncertainties linked to the complexity of the system (dumbing down), inevitably creates confusion and division among stakeholders.

Some authors have proposed lists of problematic aspects to address when studying biological invasions. Essl et al. (2017) provide a list of 13 principles that should be acknowledged to avoid such pitfalls and inform alien species management. They list uncertainty as one of these principles. Courchamp et al. (2017) outline 24 specificities and problems linked to invasion sciences, with many related to different types of uncertainty. The limited amount of time that has been available to study biological invasions suggests that there are still many uncertainties concerning such processes. Uncertainties are an integral part of any science, especially complex ones like ecology and invasion, and the field is certainly progressing toward reducing or at least acknowledging and communicating uncertainties.

Uncertainties nonetheless raise doubts in public perception, are exploited by denialists, and therefore potentially hinder the implementation of much needed management actions. Value-based subjective judgments, especially, can also lead to opposite conclusions on the impacts of alien species, and an integration of different perspectives to move toward a more mature, integrative science has been advocated (Munro et al. 2019). With the classification framework based on the four components presented here, we have (1) identified in more detail what we consider to be the key dimensions of uncertainty in invasion science; (2) explained how they impair management practices; and (3) suggested specific solutions to reduce each of them (Table 1). We hope that this list can be used as a benchmark on which invasion scientists, policy makers, and stakeholders in general can compare specific issues and identify or reduce potential uncertainties. First and foremost, we urge care with respect to the use of terminology; linguistic uncertainties are the easiest ones to avoid and are probably the ones that cast the most discredit on science. When ambiguities exist that cannot be avoided or solved in the short term for practical reasons, such as the endorsement of different definitions of invasive by different bodies, terms should be explicitly defined. Significant efforts should also be directed at acquiring high-quality (and appropriately annotated and accessible) data to minimize epistemic uncertainty (McGeoch et al. 2012). When measuring impacts, the assumptions and causal relationships between the different elements of the socio-ecological system of interest should also be clearly stated. And, since values are inherently linked to the implementation of policies for conservation issues, publications addressing such issues should be clear about the separation between science (how to measure impact with respect to a given framework) and values (the reasons for using one framework over another; Jeschke et al. 2014, Essl et al. 2017). Techniques such as scenario development are useful for making long-term plans and designing policies while accommodating different types of uncertainty, which nonetheless need to be identified. Finally, sharing practical lessons on how to deal with uncertainties between disciplines (e.g., climate change, conservation, and invasion science) will be vital if we are to successfully address the ecological and socioeconomic issues caused by global change.

Acknowledgments

Financial support was provided by the DST-NRF Centre of Excellence for Invasion Biology and the National Research Foundation of South Africa (Grant 85417 to DMR, 89967 to CH, and 86894 to JRUW). FAY also acknowledges the Working for Water Programme through the collaborative research project on “Research for Integrated Management of Invasive Alien Species.”

Literature Cited

Allsopp, M., and M. Cherry. 2004. Assessment of the economic impact on the bee and agricultural industries in the Western Cape of the clearing of certain Eucalyptus species. Agricultural Research Council, Stellenbosch, South Africa.

Angelstam, P., O. Khaulyak, T. Yamelynets, G. Mozgeris, V. Naumov, T. J. Chmielewski, M. Elbakidzea, M. Mantonaf, B. Protscg, and S. Valasiuk. 2017. Green infrastructure development at European Union’s eastern border: effects of road infrastructure and forest habitat loss. Journal of Environmental Management 193:300–311.

Bacher, S., et al. 2018. Socio-economic impact classification of alien taxa (SEICAT). Methods in Ecology and Evolution 9:159–168.

Bax, N., A. Williamson, M. Aguero, E. Gonzalez, and W. Geeves. 2003. Marine invasive alien species: a threat to global biodiversity. Marine Policy 27:313–323.

Beinart, W., and L. Wotshela. 2012. Prickly pear: a social history of a plant in the Eastern Cape. Wits University Press, Johannesburg, South Africa.

Berthouly-Salazar, C., C. Hui, T. M. Blackburn, C. Gaboriaud, B. J. Van Rensburg, B. Jansen Van Vuuren, and J. J. Le Roux. 2013. Long-distance dispersal maximizes evolutionary potential during rapid geographic range expansion. Molecular Ecology 22:5793–5804.

Blackburn, T. M., P. Pyšek, S. Bacher, J. T. Carlton, R. P. Duncan, V. Jarošík, J. R. U. Wilson, and D. M. Richardson. 2011. A proposed unified framework for biological invasions. Trends in Ecology & Evolution 26:333–339.

Blackburn, T. M., et al. 2014. A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biology 12:e1001850.

Blackburn, T. M., et al. 2015. Tools to address invasive species impacts must build on knowledge not ignorance–some thoughts on Ojaveer et al. PLoS Biology. https://journals.plos.org/plosbiology/article/comment?id=info:doi/10.1371/annotation/424f05ab-92cd-4bd4-a406-af9a4353919f

Bojinski, S., M. Verstraete, T. C. Peterson, C. Richter, A. Simmons, and M. Zemp. 2014. The concept of essential climate variables in support of climate research, applications, and policy. Bulletin of the American Meteorological Society 95:1431–1443.

Boltovskoy, D., F. Sylvester, and E. M. Paolucci. 2018. Invasive species denialism: sorting out facts, beliefs, and definitions. Ecology and Evolution 8:11190–11198.

Branch, G. M., and C. N. Steffani. 2004. Can we predict the effects of alien species? A case-history of the invasion of South Africa by *Mytilus galloprovincialis* (Lamarck). Journal of Experimental Marine Biology and Ecology 300:189–215.

Brooks, M. L. 2003. Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. Journal of Applied Ecology 40:344–353.

Brooks, M. L., C. M. D'Antonio, D. M. Richardson, J. B. Grace, J. E. Keeley, J. M. DiTomaso, R. J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. AIBS Bulletin 54:677–688.

Brooks, T. M., J. F. Lamoreux, and J. Soberón. 2014. Ipbes ≠ ipcc. Trends in Ecology & Evolution 29:543–545.

Bull, J. W., K. B. Suttle, A. Gordon, N. J. Singh, and E. J. Milner-Gulland. 2013. Biodiversity offsets in theory and practice. Oryx 47:369–380.

Cadotte, M. W., S. E. Campbell, S. P. Li, D. S. Sodhi, and N. E. Mandrak. 2018. Preadaptation and naturalization of nonnative species: Darwin's two fundamental insights into species invasion. Annual Review of Plant biology 69:661–684.

Canavan, S., S. Kumschick, J. J. Le Roux, D. M. Richardson, and J. R. U. Wilson. 2019*a*. Does origin matter for impacts of weedy plants? Not for bamboos Plants, People, Planet. https://doi.org/10.1002/ppp3.5

Canavan, S., L. A. Meyerson, J. G. Packer, P. Pyšek, N. Maurel, V. Lozano, D. M. Richardson, G. Brundu, K. Canavan, and A. Cicatelli. 2019*b*. Tall-statured grasses: a useful functional group for invasion science. Biological Invasions 21:37–58.

Cardinale, B. J., et al. 2012. Biodiversity loss and its impact on humanity. Nature 486:59.

Catford, J. A., J. B. Baumgartner, P. A. Vesk, M. White, Y. M. Buckley, and M. A. McCarthy. 2016. Disentangling the four demographic dimensions of species invasiveness. Journal of Ecology 104:1745–1758.

Catford, J. A., R. Jansson, and C. Nilsson. 2009. Reducing redundancy in invasion ecology by integrating hypotheses into a single theoretical framework. Diversity and Distributions 15:22–40.

Catford, J. A., et al. 2012. The intermediate disturbance hypothesis and plant invasions: implications for species richness and management. Perspectives in Plant Ecology, Evolution and Systematics 14:231–241.

CBD – Secretariat of the Convention on Biological Diversity. 2006. Global biodiversity outlook 2. Page 81+vii. UNO-Secretariat of the Convention on Biological Diversity, Montreal, Quebec, Canada.

CBD – Secretariat of the Convention on Biological Diversity. 2010. Decision X/2, The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets, Nagoya, Japan, 18 to 29 October 2010.

Colautti, R. I., and H. J. MacIsaac. 2004. A neutral terminology to define 'invasive' species. Diversity and Distributions 10:135–141.

Colautti, R. I., A. Ricciardi, I. A. Grigorovich, and H. J. MacIsaac. 2004. Is invasion success explained by the enemy release hypothesis? Ecology Letters 7:721–733.

Collomb, J.-D. 2014. The ideology of climate change denial in the United States. European Journal of American Studies 9:1–20.

Costanza, R., R. de Groot, L. Braat, I. Kubiszewski, L. Fioramonti, P. Sutton, S. Farber, and M. Grasso. 2017. Twenty years of ecosystem services: How far have we come and how far do we still need to go? Ecosystem Services 28:1–16.

Costanza, R., and C. Folke. 1997. Valuing ecosystem services with efficiency, fairness and sustainability as goals. *In* G. C. Daily, editor. Nature's services: societal dependence on natural ecosystems. Island Press, Washington, D.C., USA.

Costanza, R., et al. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253–260.

Courchamp, F., A. Fournier, C. Bellard, C. Bertelsmeier, E. Bonnaud, J. M. Jeschke, and J. C. Russell. 2017. Invasion biology: specific problems and possible solutions. Trends in Ecology & Evolution 32:13–22.

Crowley, S. L., S. Hinchliffe, S. M. Redpath, and R. A. McDonald. 2017. Disagreement about invasive species does not equate to denialism: a response to Russell and Blackburn. Trends in Ecology & Evolution 32:228–229.

Cullen, J. M., and E. S. Delfosse. 1984. *Echium plantagineum*: catalyst for conflict and change in Australia. Pages 19–25 *in* Proceedings of the VI International Symposium on Biological Control of Weeds. Agriculture Canada, Ottawa, Canada.

Darwin, C. 1859. On the origin of species by means of natural selection, or preservation of favoured races in the struggle for life. John Murray, London, UK.

Davis, M. A., and M. K. Chew. 2017. 'The denialists are coming!' well, not exactly: a response to Russell and Blackburn. Trends in Ecology & Evolution 32:229–230.

Davis, M. A., M. K. Chew, R. J. Hobbs, A. E. Lugo, J. J. Ewel, G. J. Vermeij, J. H. Brown, M. L. Rosenzweig, M. R. Gardener, and S. P. Carroll. 2011. Don't judge species on their origins. Nature 474:153–154.

Dickie, I. A., B. M. Bennett, L. E. Burrows, M. A. Nuñez, D. A. Peltzer, A. Porté, D. M. Richardson, M. Rejmánek, P. W. Rundel, and B. W. Van Wilgen. 2014. Conflicting values: ecosystem services and invasive tree management. Biological Invasions 16:705–719.

Dornelas, M., et al. 2018. BioTIME: a database of biodiversity time series for the Anthropocene. Global Ecology and Biogeography 27:760–786.

Downey, P. O., and D. M. Richardson. 2016. Alien plant invasions and native plant extinctions: a six-threshold framework. AoB Plants 8:plw047.

Driscoll, D. A., and M. J. Watson. 2019. Science denialism and compassionate conservation: response to Wallach et al. 2018. Conservation Biology. https://doi.org/10.1111/cobi.13273

Duffy, D. C. 2013. The games ecologists play: the contrarian. Bulletin of the Ecological Society of America 94:380.

Dyer, E. E., D. W. Redding, and T. M. Blackburn. 2017. The global avian invasions atlas, a database of alien bird distributions worldwide. Scientific Data 4:170041.

Early, R., B. A. Bradley, J. S. Dukes, J. J. Lawler, J. D. Olden, D. M. Blumenthal, P. Gonzalez, E. D. Grosholz, I. Ibañez, and L. P. Miller. 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. Nature Communications 7:12485.

Essl, F., S. Bacher, P. Genovesi, P. E. Hulme, J. M. Jeschke, S. Katsanevakis, I. Kowarik, I. Kühn, P. Pyšek, and W. Rabitsch. 2018. Which taxa are alien? Criteria, applications, and uncertainties. BioScience 68:496–509.

Essl, F., S. Dullinger, W. Rabitsch, P. E. Hulme, P. Pyšek, J. R. U. Wilson, and D. M. Richardson. 2015. Historical legacies accumulate to shape future biodiversity in an era of rapid global change. Diversity and Distributions 21:534–547.

Essl, F., et al. 2017. Scientific and normative foundations for the valuation of alien-species impacts: thirteen core principles. BioScience 67:166–178.

Evans, T., S. Kumschick, and T. M. Blackburn. 2016. Application of the Environmental Impact Classification for Alien Taxa (EICAT) to a global assessment of alien bird impacts. Diversity and Distributions 22:919–931.

Faulkner, K. T., M. P. Robertson, M. Rouget, and J. R. U. Wilson. 2016. Understanding and managing the introduction pathways of alien taxa: South Africa as a case study. Biological Invasions 18:73–87.

Floerl, O., T. K. Pool, and G. J. Inglis. 2004. Positive interactions between nonindigenous species facilitate transport by human vectors. Ecological Applications 14:1724–1736.

Forsyth, G. G., D. M. Richardson, P. J. Brown, and B. W. Van Wilgen. 2004. A rapid assessment of the invasive status of *Eucalyptus* species in two South African provinces: working for water. South African Journal of Science 100:75–77.

Gaertner, M., R. Biggs, M. Te Beest, C. Hui, J. Molofsky, and D. M. Richardson. 2014. Invasive plants as drivers of regime shifts: identifying high-priority invaders that alter feedback relationships. Diversity and Distributions 20:733–744.

García-Berthou, E., C. Alcaraz, Q. Pou-Rovira, L. Zamora, G. Coenders, and C. Feo. 2005. Introduction pathways and establishment rates of invasive aquatic species in Europe. Canadian Journal of Fisheries and Aquatic Sciences 62:453–463.

Goode, E. 2016. Invasive species aren't always unwanted. The New York Times.

Goodenough, A. 2010. Are the ecological impacts of alien species misrepresented? A review of the "native good, alien bad" philosophy. Community Ecology 11:13–21.

Graves, S. D., and A. M. Shapiro. 2003. Exotics as host plants of the California butterfly fauna. Biological Conservation 110:413–433.

Guiaşu, R. C., and C. W. Tindale. 2018. Logical fallacies and invasion biology. Biology & Philosophy 33:34.

Hagen, B. L., and S. Kumschick. 2018. The relevance of using various scoring schemes revealed by an impact assessment of feral mammals. NeoBiota 38:37–75.

Hassan, A., and A. Ricciardi. 2014. Are non-native species more likely to become pests? Influence of biogeographic origin on the impacts of freshwater organisms 3. Frontiers in Ecology and the Environment 12:218–223.

Hawkins, C. L., et al. 2015. Framework and guidelines for implementing the proposed IUCN Environmental Impact Classification for Alien Taxa (EICAT). Diversity and Distributions 21:1360–1363.

Heger, T., W.-C. Saul, and L. Trepl. 2013. What biological invasions 'are' is a matter of perspective. Journal for Nature Conservation 21:93–96.

Hobbs, R. J., et al. 2014. Managing the whole landscape: historical, hybrid, and novel ecosystems. Frontiers in Ecology and the Environment 12:557–564.

Hoffmann, B. D., and F. Courchamp. 2016. Biological invasions and natural colonisations: Are they that different? NeoBiota 29:1–14.

Hooper, D. U., E. C. Adair, B. J. Cardinale, J. E. K. Byrnes, B. A. Hungate, K. L. Matulich, A. Gonzalez, J. E. Duffy, L. Gamfeldt, and M. I. O'Connor. 2012. A global synthesis reveals biodiversity loss as a major driver of ecosystem change. Nature 486:105–108.

Hui, C., G. A. Fox, and J. Gurevitch. 2017. Scale-dependent portfolio effects explain growth inflation and volatility reduction in landscape demography. Proceedings of the National Academy of Sciences USA 114:12507–12511.

Hui, C., and D. M. Richardson. 2017. Invasion dynamics. Oxford University Press, Oxford, UK.

Hui, C., and D. M. Richardson. 2019. How to invade an ecological network. Trends in Ecology & Evolution 34:121–131.

Hui, C., D. M. Richardson, P. Pyšek, J. J. Le Roux, T. Kučera, and V. Jarošík. 2013. Increasing functional modularity with residence time in the co-distribution of native and introduced vascular plants. Nature Communications 4:2454.

Hulme, P. E. 2003. Biological invasions: Winning the science battles but losing the conservation war? Oryx 37:178–193.

Hulme, P. E. 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. Journal of Applied Ecology 46:10–18.

Hulme, P. E., and M. Bernard-Verdier. 2018. Comparing traits of native and alien plants: Can we do better? Functional Ecology 32:117–125.

Hulme, P. E., et al. 2008. Grasping at the routes of biological invasions: a framework for integrating pathways into policy. Journal of Applied Ecology 45:403–414.

IPBES. 2018. Information on scoping for a thematic assessment of invasive alien species and their control (deliverable 3 (b) (ii)). https://www.ipbes.net/system/tdf/ipbes-6-inf-10_en.pdf?file=1&type=node&id=16523

IPCC. 2014. Climate change 2014: synthesis report. *In* Core Writing Team, R. K. Pachauri, and L. A. Meyer, editors. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.

ISAC – Invasive Species Advisory Committee. 2006. Invasive species definition clarification and guidance white paper. http://www.invasivespeciesinfo.gov/docs/council/isacdef.pdf

IUCN. 2017*a*. Consultation document. IUCN standard classification of the impact of invasive alien taxa. Version 1 May 2017. https://www.iucn.org/sites/dev/files/eicat_standard_version_1_may_2017.pdf

IUCN. 2017*b*. The IUCN red list of threatened species. Version 2017-3. http://www.iucnredlist.org

Jeschke, J. M., et al. 2014. Defining the impact of non-native species. Conservation Biology 28:1188–1194.

Kareiva, P., and M. Marvier. 2012. What is conservation science? BioScience 62:962–969.

Katsanevakis, S., A. Zenetos, C. Belchior, and A. C. Cardoso. 2013. Invading European Seas: assessing pathways of introduction of marine aliens. Ocean & Coastal Management 76:64–74.

Kesner, D., and S. Kumschick. 2018. Gastropods alien to South Africa cause severe environmental harm in their global alien ranges across habitats. Ecology and Evolution 8:8273–8285.

Kruger, F. J., D. M. Richardson, and B. W. Van Wilgen. 1986. Processes of invasion by alien plants. Ecology and management of biological invasions in southern Africa: Proceedings of the National Synthesis Symposium on the Ecology of Biological Invasions. Pages 145–155 *in* I. A. W. Macdonald, F. J. Kruger, and A. A. Ferrar, editors. The ecology and management of biological invasions in South Africa. Oxford University Press, Cape Town, South Africa.

Kueffer, C., P. Pyšek, and D. M. Richardson. 2013. Integrative invasion science: model systems, multi-site studies, focused meta-analysis and invasion syndromes. New Phytologist 200:615–633.

Kumschick, S., G. J. Measey, G. Vimercati, F. A. De Villiers, M. M. Mokhatla, S. J. Davies, C. J. Thorp, A. D. Rebelo, T. M. Blackburn, and F. Kraus. 2017. How repeatable is the Environmental Impact Classification of Alien Taxa (EICAT)? Comparing independent global impact assessments of amphibians. Ecology and Evolution 7:2661–2670.

Latombe, G., D. M. Richardson, P. Pyšek, T. Kučera, and C. Hui. 2018. Drivers of species turnover of native and alien plants with different residence times vary with species commonness. Ecology 99:2763–2775.

Latombe, G., et al. 2017. A vision for global monitoring of biological invasions. Biological Conservation 213:295–308.

Le Maitre, D. C., G. G. Forsyth, S. Dzikiti, and M. B. Gush. 2016. Estimates of the impacts of invasive alien plants on water flows in South Africa. Water SA 42:659–672.

Leishman, M. R., V. P. Thomson, and J. Cooke. 2010. Native and exotic invasive plants have fundamentally similar carbon capture strategies. Journal of Ecology 98:28–42.

Levins, R., and D. Culver. 1971. Regional coexistence of species and competition between rare species. Proceedings of the National Academy of Sciences USA 68:1246–1248.

Li, F., X. Liu, X. Zhang, D. Zhao, H. Liu, C. Zhou, and R. Wang. 2017. Urban ecological infrastructure: an integrated network for ecosystem services and sustainable urban systems. Journal of Cleaner Production 163:S12–S18.

MacDougall, A. S., and R. Turkington. 2005. Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology 86:42–55.

Mace, G. M. 2014. Whose conservation? Science 345:1558–1560.

McGeoch, M. A., and G. Latombe. 2016. Characterizing common and range expanding species. Journal of Biogeography 43:217–228.

McGeoch, M. A., D. Spear, E. J. Kleynhans, and E. Marais. 2012. Uncertainty in invasive alien species listing. Ecological Applications 22:959–971.

McKenney, B. A., and J. M. Kiesecker. 2010. Policy development for biodiversity offsets: a review of offset frameworks. Environmental Management 45:165–176.

Meiners, S. J. 2007. Native and exotic plant species exhibit similar population dynamics during succession. Ecology 88:1098–1104.

Munro, D., J. Steer, and W. Linklater. 2019. On allegations of invasive species denialism. Conservation Biology. https://doi.org/10.1111/cobi.13278

Nackley, L. L., A. G. West, A. L. Skowno, and W. J. Bond. 2017. The nebulous ecology of native invasions. Trends in Ecology & Evolution 32:814–824.

Novoa, A., et al. 2018. A framework for engaging stakeholders on the management of alien species. Journal of Environmental Management 205:286–297.

Nykvist, B., and J. von Heland. 2014. Social-ecological memory as a source of general and specified resilience. Ecology and Society 19:47.

Ojaveer, H., et al. 2015. Classification of non-indigenous species based on their impacts: considerations for application in marine management. PloS Biology 13:e1002130.

Ojaveer, H., et al. 2018. Historical baselines in marine bioinvasions: implications for policy and management. PLoS ONE 13:e0202383.

Olivier, D., L. Heinecken, and S. Jackson. 2013. Mussel and oyster culture in Saldanha Bay, South Africa: potential for suitable growth, development and employment creation. Food Security 5:215–267.

Pagad, S., P. Genovesi, L. Carnevali, R. Scalera, and M. Clout. 2015. IUCN SSC Invasive Species Specialist Group: invasive alien species information management supporting practitioners, policy makers and decision takers. Management of Biological Invasions 6:127–135.

Paolucci, E. M., H. J. MacIsaac, and A. Ricciardi. 2013. Origin matters: Alien consumers inflict greater damage on prey populations than do native consumers. Diversity and Distributions 19:988–995.

Papacostas, K. J., E. W. Rielly-Carrol, S. E. Georgian, D. J. Long, S. D. Princiotta, A. M. Quattrini, K. E. Reuter, and A. L. Freestone. 2017. Biological mechanisms of marine invasions. Marine Ecology Progress Series 565:251–268.

Papworth, S. K., J. Rist, L. Coad, and E. J. Milner-Gulland. 2009. Evidence for shifting baseline syndrome in conservation. Conservation Letters 2:93–100.

Pauchard, A., et al. 2018. Biodiversity assessments: Origin matters. PLoS Biology 16:e2006686.

Pereira, H. M., et al. 2013. Essential biodiversity variables. Science 339:277–278.

Peterson, T. C., and R. S. Vose. 1997. An overview of the Global Historical Climatology Network temperature database. Bulletin of the American Meteorological Society 78:2837–2850.

Pivello, V. R., M. V. Vieira, M. T. Grombone-Guaratini, and D. M. S. Matos. 2018. Thinking about superdominant populations of native species–Examples from Brazil. Perspectives in Ecology and Conservation 16:74–82.

Pocock, M. J. O., et al. 2018. A vision for global biodiversity monitoring with citizen science. Advances in Ecological Research 59. https://doi.org/10.1016/bs.aecr.2018.06.003

Pyšek, P., V. Jarošík, and T. Kučera. 2003. Inclusion of native and alien species in temperate nature reserves: an historical study from Central Europe. Conservation Biology 17:1414–1424.

Pyšek, P., and D. M. Richardson. 2010. Invasive species, environmental change and management, and health. Annual Review of Environment and Resources 35:25–55.

Regan, H. M., M. Colyvan, and M. A. Burgman. 2002. A taxonomy and treatment of uncertainty for ecology and conservation biology. Ecological Applications 12:618–628.

Reichard, S. H., and P. White. 2001. Horticulture as a pathway of invasive plant introductions in the United States: Most invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. AIBS Bulletin 51:103–113.

Rejmánek, M., D. M. Richardson, S. I. Higgins, M. J. Pitcairn, and E. Grotkopp. 2005. Ecology of invasive plants: state of the art. Pages 104–161 *in* H. A. Mooney, R. N. Mack, J. A. McNeely, L. E. Neville, P. J. Schei, and J. K. Waage, editors. Invasive alien species: a new synthesis. Island Press, Washington, D.C., USA.

Renault, D., M. Laparie, S. J. McCauley, and D. Bonte. 2018. Environmental adaptations, ecological filtering, and dispersal central to insect invasions. Annual Review of Entomology 63:345–368.

Ricciardi, A., and R. Ryan. 2018. The exponential growth of invasive species denialism. Biological Invasions 20:549–553.

Richardson, D. M., J. J. Le Roux, and J. R. U. Wilson. 2015. Australian acacias as invasive species: lessons to be learnt from regions with long planting histories. Southern Forests: A Journal of Forest Science 77:31–39.

Richardson, D. M., and P. Pyšek. 2006. Plant invasions: merging the concepts of species invasiveness and community invasibility. Progress in Physical Geography 30:409–431.

Richardson, D. M., P. Pyšek, and J. T. Carlton. 2011. A compendium of essential concepts and terminology in invasion ecology. Pages 409–420 *in* Fifty years of invasion ecology. The legacy of Charles Elton. Wiley-Blackwell, Oxford, UK.

Richardson, D. M., P. Pyšek, M. Rejmánek, M. G. Barbour, F. D. Panetta, and C. J. West. 2000. Naturalization and invasion of alien plants: concepts and definitions. Diversity and Distributions 6:93–107.

Richardson, D. M., and A. Ricciardi. 2013. Misleading criticisms of invasion science: a field guide. Diversity and Distributions 19:1461–1467.

Robinson, T. B., M. E. Alexander, C. A. Simon, C. L. Griffiths, K. Peters, S. Sibanda, S. Miza, B. Groenewald, P. Majiedt, and K. J. Sink. 2016. Lost in translation? Standardising the terminology used in marine invasion biology and updating South African alien species lists. African Journal of Marine Science 38:129–140.

Robinson, T. B., and C. Swart. 2015. Distribution and impact of the alien anemone *Sagartia ornata* in the West Coast National Park. Koedoe 57:a1246.

Rodriguez McRobbie, L. 2016. Humans make a mess, but invasive species get the blame. The Boston Globe.

Running, S. W., D. D. Baldocchi, D. P. Turner, S. T. Gower, P. S. Bakwin, and K. A. Hibbard. 1999. A global terrestrial monitoring network integrating tower fluxes, flask sampling, ecosystem modeling and EOS satellite data. Remote Sensing of Environment 70:108–127.

Russell, J. C., and T. M. Blackburn. 2017*a*. The rise of invasive species denialism. Trends in Ecology & Evolution 32:3–6.

Russell, J. C., and T. M. Blackburn. 2017*b*. Invasive alien species: denialism, disagreement, definitions, and dialogue. Trends in Ecology & Evolution 32:312–314.

Sadchatheeswaran, S., G. M. Branch, C. L. Moloney, and T. B. Robinson. 2018. Impacts of alien ecosystem engineers overwhelm inter-annual and seasonal shifts in rocky-shore community composition on Marcus Island, South Africa. African Journal of Marine Science 40:137–147.

Sagoff, M. 2009. Environmental harm: political not biological. Journal of Agricultural and Environmental Ethics 22:81–88.

Sagoff, M. 2018. What is invasion biology? Ecological Economics 154:22–30.

Sala, O. E., F. S. Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, and A. Kinzig. 2000. Global biodiversity scenarios for the year 2100. Science 287:1770–1774.

Saul, W.-C., and J. M. Jeschke. 2015. Eco-evolutionary experience in novel species interactions. Ecology Letters 18:236–245.

Saul, W.-C., J. Jeschke, and T. Heger. 2013. The role of eco-evolutionary experience in invasion success. NeoBiota 17:57–74.

Saul, W.-C., H. E. Roy, O. Booy, L. Carnevali, H. Chen, P. Genovesi, C. A. Harrower, P. E. Hulme, S. Pagad, and J. Pergl. 2017. Assessing patterns in introduction pathways of alien species by linking major invasion data bases. Journal of Applied Ecology 54:657–669.

Scalera, R., et al. 2016. Technical report: progress towards pathways prioritization in compliance to Aichi Target 9. Information documented presented at SBSTTA 20 UNEP. CBD/SBSTTA/20/INF/5, the twentieth meeting of the CBD's Subsidiary Body on Scientific, Technical and Technological Advice, Montreal, Canada, 25–30 April 2016.

Schlaepfer, M. A. 2018. Do non-native species contribute to biodiversity? PLoS Biology 16:e2005568.

Schmeller, D. S., R. Julliard, P. J. Bellingham, M. Böhm, N. Brummitt, A. Chiarucci, D. Couvet, S. Elmendorf, D. M. Forsyth, and J. G. Moreno. 2015. Towards a global terrestrial species monitoring program. Journal for Nature Conservation 25:51–57.

Seebens, H., T. M. Blackburn, E. E. Dyer, P. Genovesi, P. E. Hulme, J. M. Jeschke, S. Pagad, P. Pyšek, M. Winter, and M. Arianoutsou. 2017. No saturation in the accumulation of alien species worldwide. Nature Communications 8:14435.

Seidensticker, J. (2008) Ecological and intellectual baselines: saving lions, tigers, and rhinos in Asia. Pages 98–117 *in* L. Rockwood, R. Stewart, and T. Dietz, editors. Foundations of environmental sustainability: the coevolution of science and policy. Oxford University Press, New York, New York, USA.

Shackleton, S., D. Kirby, and J. Gambiza. 2011. Invasive plants–friends or foes? Contribution of prickly pear (*Opuntia ficus-indica*) to livelihoods in Makana Municipality, Eastern Cape, South Africa. Development Southern Africa 28:177–193.

Shackleton, C. M., D. McGarry, S. Fourie, J. Gambiza, S. E. Shackleton, and C. Fabricius. 2007. Assessing the effects of invasive alien species on rural livelihoods: case examples and a framework from South Africa. Human Ecology 35:113–127.

Shackleton, R. T., D. M. Richardson, C. M. Shackleton, B. Bennett, S. L. Crowley, K. Dehnen-Schmutz, R. A. Estévez, A. Fischer, C. Kueffer, and C. A. Kull. 2019. Explaining people's perceptions of invasive alien species: a conceptual framework. Journal of Environmental Management 229:10–26.

Simberloff, D., J.-L. Martin, P. Genovesi, V. Maris, D. A. Wardle, J. Aronson, F. Courchamp, B. Galil, E. García-Berthou, and M. Pascal. 2013. Impacts of biological invasions: what's what and the way forward. Trends in Ecology & Evolution 28:58–66.

Simberloff, D., L. Souza, M. A. Nuñez, M. N. Barrios-Garcia, and W. Bunn. 2012. The natives are restless, but not often and mostly when disturbed. Ecology 93:598–607.

Soga, M., and K. J. Gaston. 2018. Shifting baseline syndrome: causes, consequences, and implications. Frontiers in Ecology and the Environment 16:222–230.

Sogge, M. K., S. J. Sferra, and E. H. Paxton. 2008. *Tamarix* as habitat for birds: implications for riparian restoration in the southwestern United States. Restoration Ecology 16:146–154.

Sorte, C. J. B., S. L. Williams, and J. T. Carlton. 2010. Marine range shifts and species introductions: comparative spread rates and community impacts. Global Ecology and Biogeography 19:303–316.

Soule, M. 2014. The 'new conservation'. Conservation Biology 27:895–897.

Spellerberg, I. A. N. 1998. Ecological effects of roads and traffic: a literature review. Global Ecology & Biogeography Letters 7:317–333.

Tassin, J., K. Thompson, S. P. Carroll, and C. D. Thomas. 2017. Determining whether the impacts of introduced species are negative cannot be based solely on science: a response to Russell and Blackburn. Trends in Ecology & Evolution 32:230–231.

Taylor, K. T., B. D. Maxwell, A. Pauchard, M. A. Nuñez, and L. J. Rew. 2016. Native versus non-native invasions: similarities and differences in the biodiversity impacts of *Pinus contorta* in introduced and native ranges. Diversity and Distributions 22:578–588.

Tecco, P. A., S. Díaz, M. Cabido, and C. Urcelay. 2010. Functional traits of alien plants across contrasting climatic and land-use regimes: Do aliens join the locals or try harder than them? Journal of Ecology 98:17–27.

Tererai, F., M. Gaertner, S. M. Jacobs, and D. M. Richardson. 2013. *Eucalyptus* invasions in riparian forests: effects on native vegetation community diversity, stand structure and composition. Forest Ecology and Management 297:84–93.

Thompson, K., J. G. Hodgson, and T. C. G. Rich. 1995. Native and alien invasive plants: More of the same? Ecography 18:390–402.

United Nations. 2015. Paris Agreement. https://unfccc.int/sites/default/files/english_paris_agreement.pdf

Valéry, L., H. Fritz, and J. Lefeuvre. 2013. Another call for the end of invasion biology. Oikos 122:1143–1146.

van Kleunen, M., W. Dawson, D. Schlaepfer, J. M. Jeschke, and M. Fischer. 2010. Are invaders different? A conceptual framework of comparative approaches for assessing determinants of invasiveness. Ecology Letters 13:947–958.

van Wilgen, N. J., M. J. Gillespie, D. M. Richardson, and J. Measey. 2018. A taxonomically and geographically constrained information base limits non-native reptile and amphibian risk assessment: a systematic review. PeerJ 6:e5850.

van Wilgen, B. W., and D. M. Richardson. 2014. Managing invasive alien trees: challenges and trade-offs. Biological Invasions 16:721–734.

van Wilgen, B., D. Richardson, and S. Higgins. 2000. Integrated control of invasive alien plants in terrestrial ecosystems. Pages 118–128 *in* Best Management Practices for Preventing and Controlling Invasive Alien Species, Symposium Proceedings. The Working for Water Programme, Cape Town, South Africa.

Vaz, A. S., C. Kueffer, C. A. Kull, D. M. Richardson, S. Schindler, A. J. Muñoz-Pajares, J. R. Vicente, J. Martins, C. Hui, and I. Kühn. 2017. The progress of interdisciplinarity in invasion science. Ambio 46:428–442.

Vilà, M., et al. 2010. How well do we understand the impacts of alien species on ecosystem services? A pan-European cross-taxa assessment. Frontiers in Ecology and the Environment 8:135–144.

Wallach, A. D., M. Bekoff, C. Batavia, M. P. Nelson, and D. Ramp. 2018. Summoning compassion to address the challenges of conservation. Conservation Biology 32:1255–1265.

Williams, S. L., et al. 2013. Managing multiple vectors for marine invasions in an increasingly connected world. BioScience 63:952–966.

Williamson, M. 1996. Biological invasions. Springer Science & Business Media, Berlin, Germany.

Wilson, J. R. U., E. E. Dormontt, P. J. Prentis, A. J. Lowe, and D. M. Richardson. 2009. Something in the way you move: Dispersal pathways affect invasion success. Trends in Ecology & Evolution 24:136–144.

Wilson, J. R. U., K. T. Faulkner, S. J. Rahlao, D. M. Richardson, T. A. Zengeya, and B. W. van Wilgen. 2018. Indicators for monitoring biological invasions at a national level. Journal of Applied Ecology 55:2612–2620.

Wilson, J. R. U., P. García-Díaz, P. Cassey, D. M. Richardson, P. Pyšek, and T. M. Blackburn. 2016. Biological invasions and natural colonisations are different–the need for invasion science. NeoBiota 31:87–98.

Wilson, J. R. U., D. M. Richardson, M. Rouget, Ş. Procheş, M. A. Amis, L. Henderson, and W. Thuiller. 2007. Residence time and potential range: crucial considerations in modelling plant invasions. Diversity and Distributions 13:11–22.

Wilson, J. R. U., et al. 2014. A standardized set of metrics to assess and monitor tree invasions. Biological Invasions 16:535–551.

Wittenberg, R., and M. J. W. Cock. 2001. Invasive alien species: a toolkit of best prevention and management practices. CAB International, Wallingford, UK.

Woodford, D. J., D. M. Richardson, H. J. MacIsaac, N. E. Mandrak, B. W. Van Wilgen, J. R. U. Wilson, and O. L. F. Weyl. 2016. Confronting the wicked problem of managing biological invasions. NeoBiota 31:63–86.

World Wide Fund for Nature. 2018. Living Planet Report – 2018: Aiming Higher. M. Grooten and R. E. A. Almond, editors. WWF, Gland, Switzerland.

Worm, B., E. B. Barbier, N. Beaumont, J. E. Duffy, C. Folke, B. S. Halpern, J. B. C. Jackson, H. K. Lotze, F. Micheli, and S. R. Palumbi. 2006. Impacts of biodiversity loss on ocean ecosystem services. Science 314:787–790.

Zengeya, T., P. Ivey, D. J. Woodford, O. Weyl, A. Novoa, R. Shackleton, D. Richardson, and B. van Wilgen. 2017. Managing conflict-generating invasive species in South Africa: challenges and trade-offs. Bothalia: African Biodiversity & Conservation 47:1–11.